Chapter 2
Principles of Life Cycle Inventory Modeling: The Basic Model, Extensions, and Conventions

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Abstract The basic model of a life cycle inventory (LCI), with unit processes as smallest modeling entities, emerged already in the very early phases of life cycle assessment (LCA) method development. It is a rather simple, linear model, with a distinction between elementary flows, product flows, and waste flows. Since the early applications, this simple model proved to be very useful and allowed for various expansions. For certain issues related to LCI modeling, solutions and approaches have evolved as extensions of the basic model. Such issues and related modeling challenges include: the multifunctionality problem; the modeling of loops in product systems; the modeling of the use phase; the modeling of transport services; the consideration of time and long-term emissions in LCI; the definition of the boundary between the technosphere and biosphere; and how to address accidents, incidents, and risks. This chapter presents and explains the basic LCA model and its extensions, where some are commonly used in practice today, and some others not. Furthermore, conventions regarding the modeling of transport services, use phase and products, end of life, are presented.

Keywords Accidents · Biogenic carbon · Biosphere · ISO standards · Life cycle inventory (LCI) model · Multifunctionality · Risks · Technosphere · Use phase

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1 The Basic Life Cycle Inventory Model

The concept of a life cycle is at the core of life cycle assessment (LCA) and of life cycle inventory (LCI) models, which, in the end, aim at modeling impacts of anthropogenic, "man-made" product systems to the environment. The impacts modeled are those linked to one product, which is followed from cradle to grave, that is, from resource extraction to its end of life. The interventions of this anthropogenic system are basically inputs and outputs, as shown in Fig. 2.1. "The environment" can be understood here in a wider sense than merely as environmental impacts, covering everything that is around the anthropogenic system investigated. It is convenient to classify the inputs and outputs of the system further. Inputs can be categorized into resources and other inputs. Outputs can be categorized into emissions to different environmental compartments (air, water, and soil), waste, and products, which are not released to the environment but used by consumers or further processed within the anthropogenic system (Fig. 2.2).

All these inputs and outputs are summarized under the term *flows*. The *product* is one of the flows; it plays a central role since it represents the benefit delivered by the system and can be seen as the reason why the system exists at all. Demand for the product created in the system triggers the resource needs and the emissions of the system. The anthropogenic system is often referred to as the *technosphere*, and its surrounding is referred to as the *biosphere* (Milsum 1968).

A *life cycle* is commonly divided into several life cycle stages, such as raw material extraction (or acquisition), manufacturing and production, distribution and transport, use and maintenance, as well as finally recycling and treatment of waste (Fig. 2.3). Considering the whole life cycle is often mentioned as key to avoid burden-shifting and to evaluate a product in a comprehensive manner (ISO 2006a; Bjørn et al. 2018), and it is at the core of the LCI modeling, as well as LCA as a whole.

The life cycle is built from, and consists of, processes that are linked by exchanging products. This means that products are delivered from one process to the next, while causing emissions and contributing to resource extraction. The product or products of the entire life cycle is/are described in terms of a functional unit. It is common that the same product from the same process occurs as an input (and/or output) several times in a life cycle, for example, electricity and transport. Figure 2.4 shows this schematically, with the product system being incomplete due to potential

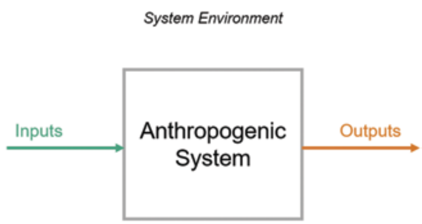

Fig. 2.1 Most basic model of life cycle inventory modeling: an anthropogenic system with interventions to the environment, as inputs and outputs. (Fava et al. 1991, adapted)

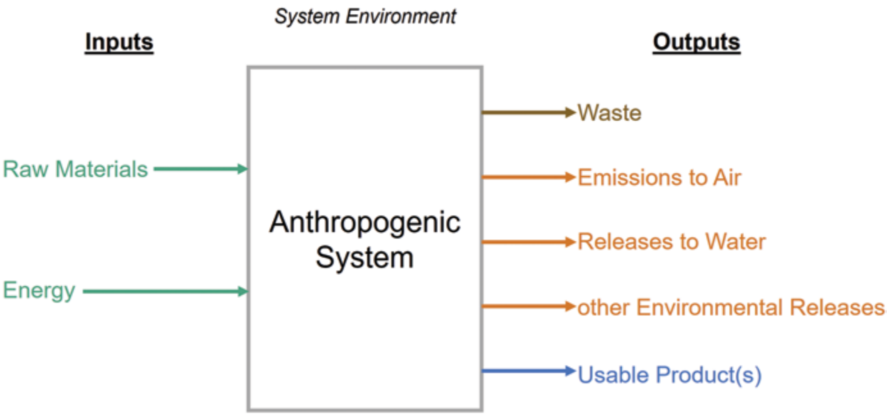

Fig. 2.2 Basic model of life cycle inventory modeling: an anthropogenic system with interventions to the environment, distinguished in different types of flows. (Fava et al. 1991, adapted)

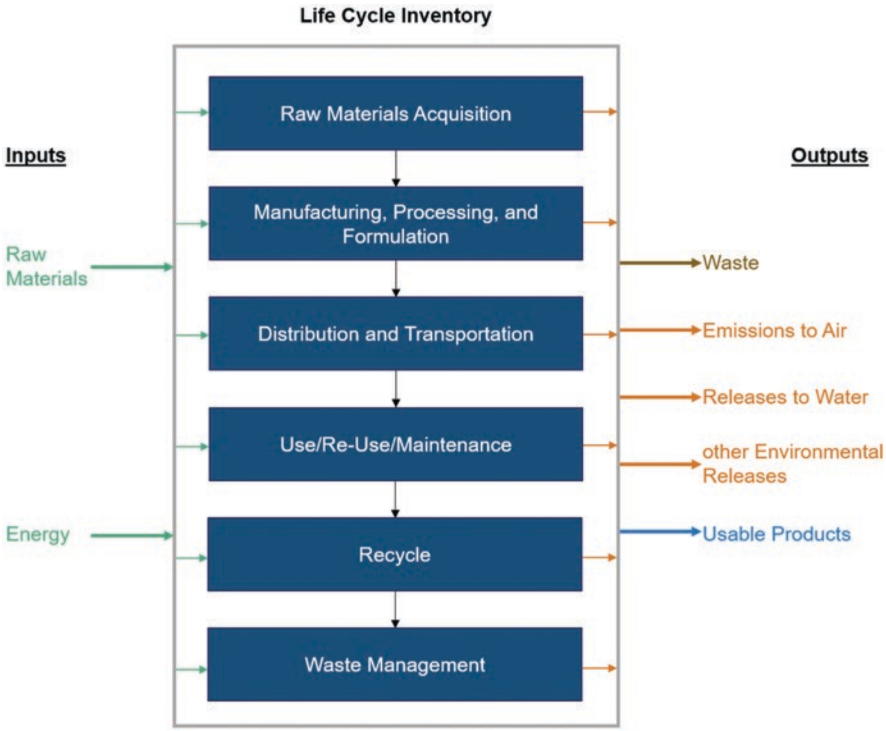

Fig. 2.3 Life cycle stages in an LCI model. (Adapted from Fava et al. 1991)

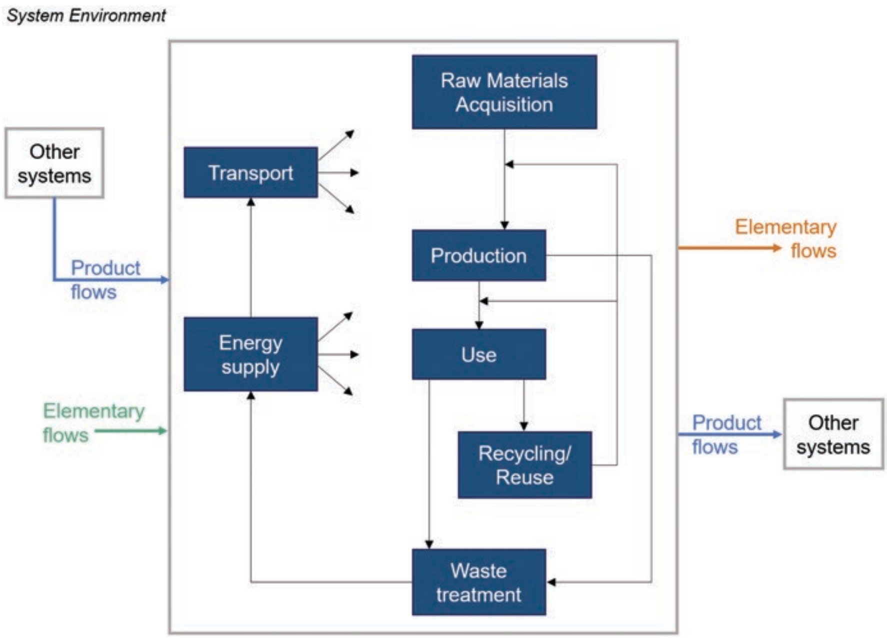

Fig. 2.4 A schematic life cycle with example processes (ISO 2006a): some flows (e.g., from transport and energy processes) can be used several times, while others can create loops within the analyzed system (e.g., flows from recycling)

inputs from other product systems in the form of products. Figure 2.5 shows parts of a more realistic case.

This bundle of connected processes is also called a *product system*. It is delivering the functional unit of an LCA case study. Each process is called *unit process* and it is typically modeled as a black box (Fig. 2.6), with a fixed relation between inputs and outputs. This means that all inputs and outputs linearly depend on the amount of the product needed (i.e., *usable product* in Fig. 2.6): if two units of product are needed instead of one, all flow amounts in the process are multiplied by a factor of two. The whole product system model in an LCA is therefore a relatively simple, but large, linear model.

For a given process, not all flow types may be present. As an example, Fig. 2.7 shows the data set of the process of soy biodiesel production from LCA Commons (https://www.lcacommons.gov/), with several input products, water as input resource, two output products (glycerin and soy biodiesel), and fatty acids as emissions into water.

This linear model, which describes processes as input/output "boxes" connected by exchanging products and distinguished into several life cycle stages, is **the basic LCI model**. As mentioned, this model is a simplification of reality in several aspects. This makes sense, since the task, to model the impacts caused by a product over its entire life cycle, is complex and demanding. In reality, life cycles are infinite

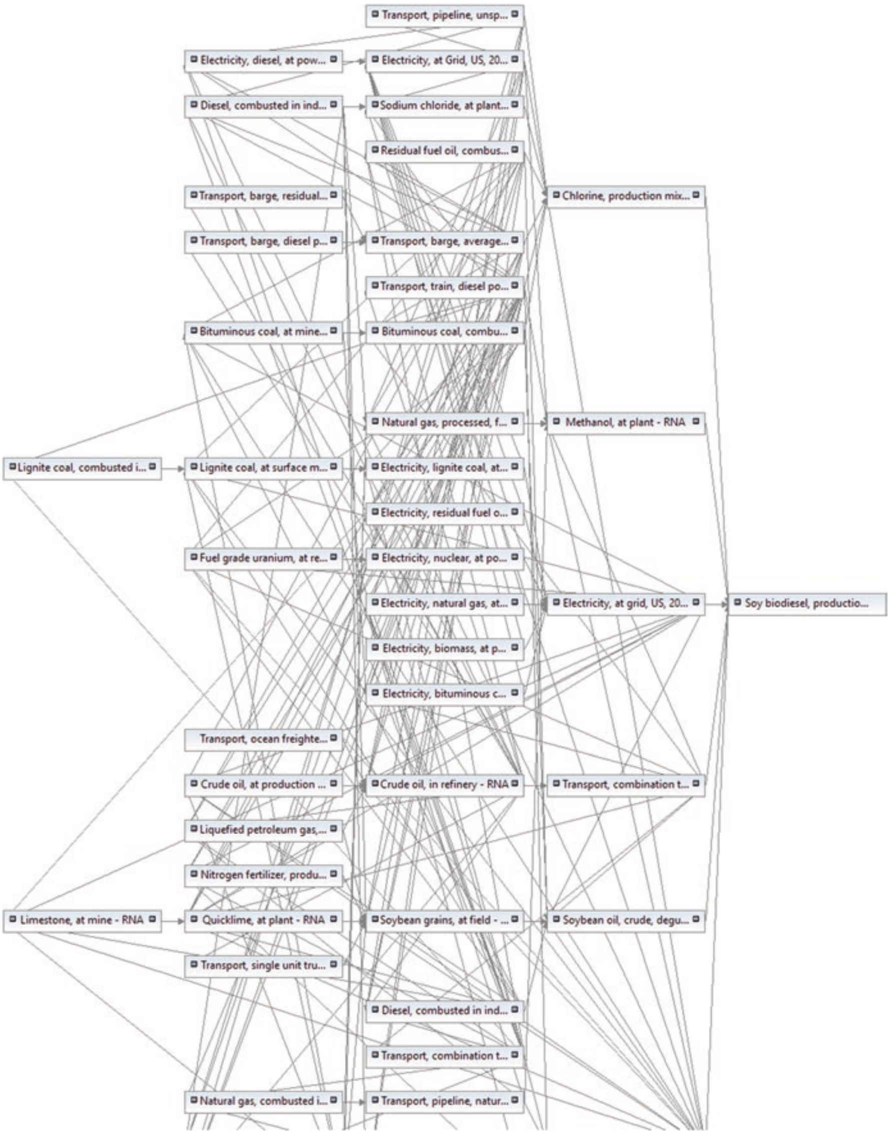

Fig. 2.5 Parts of a realistic life cycle for soy biodiesel production. Screenshot from openLCA using processes (represented through boxes) from LCA Commons (https://www.lcacommons.gov/)

and for many processes, the input amounts do not depend linearly on the product produced (e.g., for agricultural processes, Heady 1958), the scale of the process has an influence on process inputs and outputs, especially for industrial processes (Encyclopaedia Britannica 2011; Piccinno et al. 2016), and impacts vary over time and space. Therefore, the basic LCI model is simple, robust, and relatively easy to

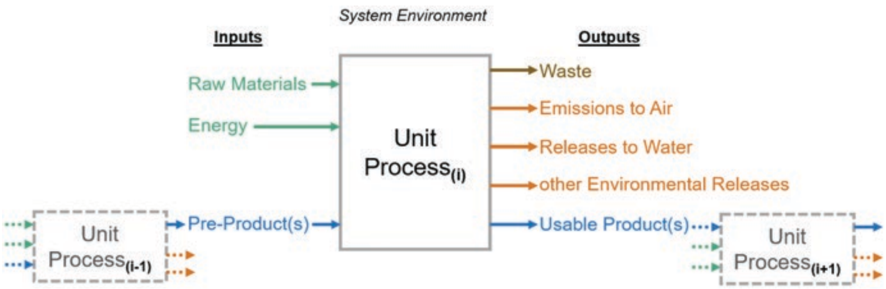

Fig. 2.6 Principal structure of a unit process data set, with energy, raw materials, and pre-products on the input side, and emissions to water, air, and other environmental compartments, as well as waste and products

Inputs				
Flow	**Category**	**Flow property**	**Unit**	**Amount**
Citric Acid, at plant		Mass	kg	0.00245
Hydrochloric Acid, at plant		Mass	kg	0.146
Phosphoric Acid, at plant		Mass	kg	0.00213
Sodium Methylate, at plant		Mass	kg	0.0777
Electricity, at grid, US, 2000	Utilities/Electric Power Distribution	Energy	kWh	0.12
Methanol, at plant	Chemical Manufacturing/All Other Basic Organic Chemical Manufacturing	Mass	kg	0.305
Natural gas, combusted in industrial boiler	Utilities/Steam and Air-Conditioning Supply	Volume	m3	0.0762
Sodium hydroxide, production mix, at plant	Chemical Manufacturing/All Other Basic Inorganic Chemical Manufacturing	Mass	kg	0.00327
Soybean oil, crude, degummed, at plant	Chemical Manufacturing/All Other Basic Organic Chemical Manufacturing	Mass	kg	3.32
Transport, combination truck, diesel powered	Truck Transportation/General Freight Trucking	Goods transport (mass*distance)	t*km	1.24
Water	resource/unspecified	Volume	l	1.14
Outputs				
Flow	**Category**	**Flow property**	**Unit**	**Amount**
Fatty acids	water/unspecified	Mass	kg	0.00694
Glycerin, at biodiesel plant	Biofuels Manufacturing/Biodiesel	Mass	kg	0.403
Soy biodiesel, production, at plant	Biofuels Manufacturing/Biodiesel	Mass	kg	3.36

Fig. 2.7 Snapshot of soy biodiesel production dataset from the LCA Commons database (https://www.lcacommons.gov/)

compute, but it presents some key modeling aspects, as well as some possibilities for extensions. In the following Sects. 2 and 3, these modeling aspects and extensions, respectively, will be described together with a number of examples. In Sect. 4, some conventions for modeling transport services, the use phase, and the end of life in LCA are presented.

2 Some Fundamental Modeling Topics in the Basic LCI Model

Some aspects are fundamental to modeling the life cycle, following the idea of a basic LCI model: modeling the benefit a product provides, setting the system boundaries, and modeling what is caused by consuming a product, that is, which other production processes are triggered if product of one specific process is consumed. Further, the location of emissions and of the activity itself, and not the least the question of when a process can be considered as being complete. These aspects are further explored in the following.

2.1 Modeling Benefits and Impacts: The Functional Unit

Already in the very early days of LCA, the emphasis was put on how to develop an inventory model (and LCA) that is suited for a fair comparison of different products. The basic idea of enabling a fair comparison is based on benefits and impacts related to a product: Every product has more or less negative impacts (emissions to the environment and resource use); on the other hand, every product brings benefits to its user, and these benefits are the reason why the product is produced and then consumed. To allow for a fair comparison of different products, it must therefore be ensured that the products to be compared provide the same benefit. If this is the case, the product with the least environmental impact is preferable. This is shown in Fig. 2.8 for a theoretical case, with three different products having different benefits and impacts. Since product 1 and product 3 bring the same benefits, it is possible to say that product 1 is better than 3 since it causes lower impacts. However, since benefits differ between product 2 and product 3, which of these is preferable is unknown – product 2 has lower impacts but also less benefits, while product 3 has more benefits but also higher impacts.

The product's benefit is represented by the *functional unit*, which is defined in ISO 14040 as "quantified performance of a product system for use as a reference unit" (ISO 2006a). The functional unit is directly linked to one specific or, more rarely, to different processes and their products via the *reference flow*, which is defined as a "*measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit*" (ISO 2006a).

In the example of the soy biodiesel process shown in Fig. 2.5, the functional unit could be: *Production of 1 liter of diesel for use in common, unmodified diesel engines, in mixture with fossil diesel, cetane number to measure ignition speed of 51* (Knothe 2006). The reference flow would be 1 liter of soy biodiesel according to the specification provided in the functional unit. With this functional unit, the soy biodiesel process and the preceding/upstream life cycle can be compared to a

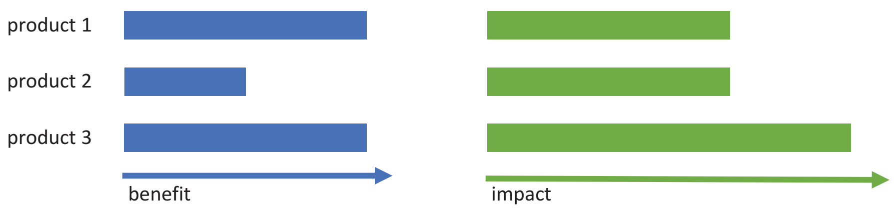

Fig. 2.8 Products 1, 2, and 3; product 2 offers less benefit than products 1 and 3, but similar or higher impact. Among products 1 and 3, product 1 has the lowest impacts and is therefore preferable

conventional diesel generated from fossil sources, with the same functional unit and an equivalent reference flow. Defining the functional unit is crucial for the outcome of a product comparison (e.g., Ciroth and Srocka (2008)), and is often discussed in LCI modeling and review. Common functional units are units of mass (1 kg or ton), volume (1 liter or m^3), energy (1 MJ or kWh), and 1 item, but also 1 km of line of writing (e.g., to assess the performance of a pen), a specific performance in insulation expressed in terms of transmittance (e.g., in $Wm^{-2}\ K^{-1}$ to assess insulation panels).

The functional unit is defined in the goal and scope phase of an LCA and represents the starting point of an inventory model (Curran 2017). This latter starts with questions as of how, meaning via which processes, the functional unit is provided or can be provided. The definition of functional unit in the goal and scope definition phase makes the inventory modeling focusing only on negative impacts related to input and output flows of processes. Only in cases of avoided or consumed emissions, and avoided resource use, can positive impacts occur. The rather unusual cases where a product has direct environmental benefit, not only indirectly via avoided emissions, are seldom considered. As examples, off-shore wind energy parks are said to have positive impacts on marine wildlife, once installed (Slavik et al. 2018); in social LCA, which generally claims to follow LCI modeling, some impacts can be positive, such as the creation of knowledge-intensive jobs and contribution to local development (Di Cesare et al. 2018). Such positive impacts are commonly not included when defining the functional unit.

2.2 *Modeling Causality: Attributional Versus Consequential Perspectives*

As explained in the previous section, an LCI model refers to the consumption of a certain amount of a given product, which is specified by the functional unit and the reference flow. The entire LCI model can then be seen as an answer to the question: Which production processes are triggered by the consumption, in the given amount, for the specified product? The answer to this question is decisive for the development of the entire model. It not only needs an answer for the final process, which is providing the product of the functional unit, but for any other product that is appearing as input in the included processes as well. In LCA, two principally different approaches for answering this question have evolved, called attributional and consequential LCA modeling. The corresponding LCA models are quite different, for example, regarding system boundaries, required input data, and allocation (Ekvall et al. 2016) (Table 2.1).

Attributional LCA was originally proposed by Heijungs (1997) with the aim of providing information on the portion of "global burden" associated with a product and its life cycle. This approach is based on the ceteris paribus assumption, meaning that the choice of the functional unit (i.e., linked to a certain amount of product)

Table 2.1 Characteristics of accounting type and change-oriented LCI models (Tillman 2000)

Characteristics	Type of LCA	
	Accounting (attributional)	Change-oriented (consequential)
System boundaries	Additivity	Part of system affected
	Completeness	
Allocation procedure	Reflecting causes of system	Reflecting effects of change
	Partitioning	System enlargement
Choice of data	Average	Marginal (at least in part)

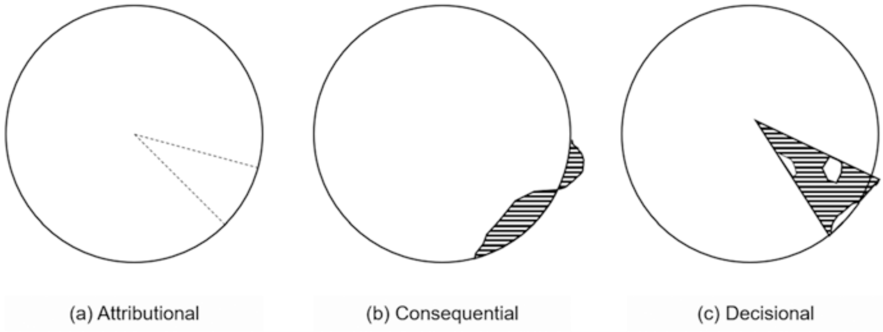

Fig. 2.9 Attributional, consequential, and decisional LCA (adapted from Weidema 2003). The entire pie represents the whole production (or market) of a product, the slice represents the functional unit analyzed in a study and the dashed areas represent the portion of the market influenced by a decision

does not influence the other activities on the planet, including the overall production and consumption of the product under study (Heijungs et al. 1992; Frischknecht 1998; Ciroth and Srocka 2008). Therefore, the functional unit can be directly linked to the overall product production, with its inputs and outputs occurring within a certain time (Curran et al. 2005), through the reference flow (i.e., amount of product required in the functional unit). Referring to Fig. 2.9a, an attributional LCA can be visualized as a slice of the whole product pie, where the functional unit defines how large the slice is.

Regarding the model, attributional LCA fits well with a linear modeling approach. If attributional LCAs of all final products were conducted, the total environmental burdens worldwide would be estimated. According to the superposition principle of linear systems, the impact of a larger system results from the sum of impacts of all single sub-systems. In addition, the resulting impacts are linearly proportional to the assumed produced quantity, and all inputs and outputs are equally allocated to each single unit of the reference product (e.g., each kg). Such an attributional analysis of a product usually considers current average market conditions.

Despite attributional LCA probably being the historically most common approach, *consequential LCA* has been gaining popularity in the last decade (McManus and Taylor 2015). Consequential modeling aims at modeling direct and indirect changes in the production system and system environment induced by decisions, or in short, consequences of decisions (Ekvall and Weidema 2004). The consequential modeling approach was proposed with the idea that attributional LCA is for certain markets not able to adequately reflect real impacts: If markets are constrained so that additional product consumption cannot be realized by increased average production, attributing the average production to new consumption is not a good estimate. In that case, no additional consumption can take place and, typically via higher prices, an increased competition for the existing products occurs, where previous consumption is discontinued. Alternatively, the market is expanded by installing new production capacity, which might be different from the market average production. These decisions can regard investments in new products or modifications of existing production processes, and can lead to changes in the market and consumption patterns, such as technology switches, changes in the market share and learning curves (Curran et al. 2005). Consequential LCA thus describes environmentally-relevant physical flows to and from a life cycle and its sub-systems that are influenced by a decision (Ekvall and Weidema 2004). These environmentally relevant consequences depend on:

1. The consumed amount of the product under study, or preferably the increased or decreased demand for the product. These changes can be short term (e.g., changes in output from existing production capacity) or long term, which regard changes in the timing, and perhaps the nature, of investments in new production capacity (Curran et al. 2005).
2. How constrained the market of this product is regarding whether additional consumption can be satisfied with the existing production capacity. Therefore, consequential modeling generally assumes that the required additional production capacity is satisfied through alternative, usually newer, technology, compared to the market average.

Referring to Fig. 2.9b, a consequential LCA can be visualized as a change to the overall product pie. The operating assumption behind consequential LCA is that a particular decision regarding a production process affects other production processes (e.g., changes in their outputs) due to cause-effect chain relationships. Specifically, the consequential approach aims to link microeconomic actions with macroeconomic consequences (Frischknecht and Stucki 2010), and it is argued to be suitable to evaluate the environmental consequences of decisions (Tillman 2000), especially of "big decisions" (Brandão et al. 2014). For this reason, the rules used to define which processes are included in or excluded from the product system are based on estimations of how material and energy flows will change due to the analyzed potential decisions, meaning that a consequential LCA study only includes processes that are affected by this decision (Curran et al. 2005). Consequently, consequential LCI models describe supply chains embedded in a dynamic technosphere that reacts to changes in the demand for different products (Sonnemann et al. 2013). Therefore, consequential LCIs could include alternative use of constrained production factors (i.e., constrained market), general market effects, identification of the competing products, identification of marginal technology, and technology development (Ekvall and Weidema 2004).

LCI models are in principle steady-state, linear, and homogeneous, with each unit process fixed at a specific point in time (Suh and Huppes 2005; Consequential-LCA 2015). But in consequential modeling, the decision depends on the specific product analyzed and amounts affected, as well as on available technologies on the market, which involves several competing alternative technologies, making the system modeling more complex. For this reason, consequential LCI models can, for example, involve partial or general economic equilibrium models to describe market reactions (Ibenholt 2002), agent-based models to include human behavior and local variabilities (Baustert and Benetto 2017), or dynamic models to build one or several scenarios to be used as (per-defined) conditions (Frischknecht and Stucki 2010). Despite consequential models are claimed to describe how activities influence each other and their environment (Weidema 2016) and thus not to be scenario modeling, they often include scenarios to describe alternative decisions (e.g., Yang 2016). While theoretically convincing, it is often difficult to model consequences in a clear and unambiguous way: Even if consequences are defined for the specific case study, a specific new technology introduced in the market can substitute several existing ones.

A third approach, called *decisional* or decision-oriented LCA (Fig 2.9c), represents an alternative definition of the consequential approach focusing on the microeconomic level (Frischknecht 1998). The main basis of information for constructing the product system in the LCI are actual or anticipated financial and contractual relations between economic actors (business-to-business relations). Consequently, the economic and/or contractual links define which processes are included or not in the LCI model (Frischknecht and Stucki 2010). Decisional LCA aims at supporting decisions in companies to improve the environmental performance of their products or processes.

This section represents a brief summary of the different LCI modeling approaches. Given the huge amount of existing literature and the ongoing debate, the presentation and discussion on the different LCI modeling approaches would have deserved a full dedicated chapter. Nevertheless, the reader can access the cited literature to dig deeper into the topic.

2.3 Setting Boundaries in an Infinite Inventory Model

Setting boundaries is one of the crucial steps in LCA and LCI modeling, since it prevents, ideally, the following of supply chains that are not contributing considerably to the overall result, and thus helps to focus on the important parts. An explicit specification of system boundary setting is fundamental also for carrying out fair comparisons of products. Issues related to setting the boundaries are present both within the technosphere as well as between the technosphere and the biosphere. Within the technosphere, any life cycle is in principle infinite (Baumann and Tillman

2004): for instance, referring to the soy biodiesel production in the US (see Fig. 2.5), it may need Japanese machines, which in turn need electricity from the Japanese electricity grid mix, which again needs, for the nuclear power share, uranium from Kazakhstan, which again is produced using machines from Russia, and so on. In order to deal with boundaries within the technosphere, two approaches are common.

Firstly, quantitative cut-off rules are applied to define the technosphere's boundaries and the related data collection. In a well-specified system, amounts of flows, scaled to the functional unit and reference flow, will become smaller and smaller while going backward in the supply chain. Also, loops in the system, created by a process delivering its inputs as outputs, directly or via other processes, will converge. For example, steel production will require some steel, but not more steel than the steel production produces. Similarly, corn production will require a certain amount of corn to be used as seed, but less than the corn that is produced, otherwise the entire production process does not make sense. Quantitative cut-off rules specify a threshold; products that do not provide, scaled to the quantitative reference flow, a certain quantitative amount above the threshold are not included in the system, nor their upstream supply chain. With amounts becoming smaller and smaller for processes that are "more remote" from the process delivering the functional unit, a threshold thus delimits the size of the overall investigated system (Fig. 2.10).

The threshold amount is typically defined based on the amount of the product in terms of energy or material. If products have different units (e.g., a liter of diesel vs kg of soybean, or a piece of a car vs amount of metal sheets in kg), the quantitative cut-off may not be able to fully reflect the quantitative contribution of the flows to the overall inventory result.

Secondly, each flow amount is only a proxy of the contribution to the environmental impacts. Therefore, the ISO 14040 standard mentions that cut-off thresholds can also be specified in terms of (relative contribution to the) environmental impacts (ISO 2006a). However, when building a product system and when deciding whether to include a new process or not, the entire environmental impact of the supply chain

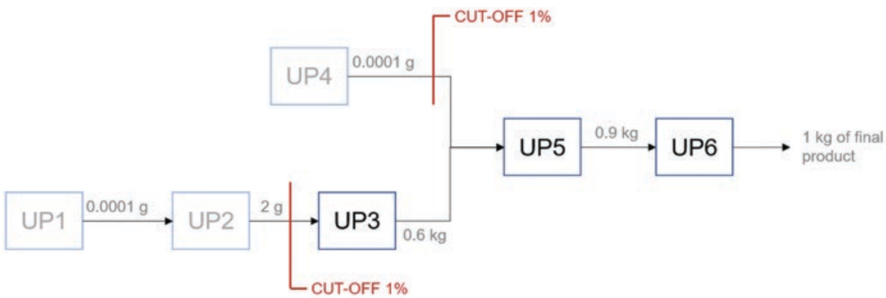

Fig. 2.10 The application of cut-off excludes the processes and the flows that contribute less than the fixed cut-off threshold. In the picture, the cut-off is set at 1% and all the flows below this threshold, that is, below 10 g, are excluded from the LCI model. (UP = unit process)

of that process are not known, and thus the environmental impact cannot really be considered as a threshold.

A third type of cut-off rules exclude processes or flows that fall into a certain type or classification, typically infrastructure. This "discriminating cut-off" is justified by analysis that under certain conditions, infrastructure does typically not contribute to the result of an LCA considerably (Frischknecht et al. 2007).

Fourthly, an LCA study can focus on some specific steps in the life cycle only, cutting off the others. Common cases are the cradle to gate studies, comprising the life cycle until the product leaves the producer, thus excluding use and end of life phase. This is, however, not typically considered as a cut-off, and will not be discussed further here.

Cut-off criteria discussed so far are applied when building the life cycle inventory. Their use must be specified in the goal and scope definition phase of the LCA. While being helpful for creating models and focusing the effort on the parts in the life cycle that matter, they are only supported directly by few LCA software systems, suffer from different units in databases,[1] and only approximating the actual impact to be assessed.

Another use of the cut-off criterion is ex post, for quality assurance of already created systems. Several Environmental Product Declarations, and also the Environmental Footprint Category Rules, specify that overall the amount of excluded materials must not exceed a threshold, or also that the excluded environmental impact must not exceed a certain threshold (e.g., European Commission 2018). In these cases, the threshold exceedance can only be calculated once the full, infinite but converging, system is known.

In the best case, no threshold and cut-off are applied. According to ISO 14040, "[t]he system boundary defines the unit processes to be included in the system. Ideally, the product system should be modeled in such a manner that inputs and outputs at its boundary are elementary flows…" Setting boundaries between the technosphere and the biosphere, that is, defining which are technical and elementary flows, is not always trivial. Potentially, this boundary can be precisely defined for nonrenewable resources (e.g., where a metal is extracted from a mine), while it is more difficult for renewable ones, both for found (e.g., forests and agricultural land) and flowing resources (e.g., solar radiation) (Baumann and Tillman 2004). Similarly, it is not always easy to define whether agricultural soils are part of the technosphere or the biosphere. Soils are essential components of technical activities such as plowing, tillage, mechanized planting, and harvesting, as well as fertilization and pest control. Nevertheless, soils have important ecological functions through which they provide the so-called supporting ecosystem services, which are part of the biosphere (MEA 2005). The definition of system boundaries in agricultural production systems is important and has a great influence of the results (Roer et al. 2012). This definition is further affected by complex soil dynamics (e.g., soil erosion, nutrient

[1] Which means, on the other hand, that they are more powerful in databases such as input-output databases where all product flows have the same unit.

leaching, nitrous oxide emissions) (Li et al. 2007), which are difficult to control. For example, regarding climate change, carbon net sequestration or emissions only occur when the soil management type has been changed until a new equilibrium level of carbon in the soil is reached (Tuomisto et al. 2012); additionally, the status of carbon in the soil and carbon flows also depend on the history and on the location of the soil under study.

The definition of boundaries between the technosphere and the biosphere regards also the “grave” side of the product life cycle. In particular, this applies to the case of landfill sites and the related long-term emissions, which are introduced in Sect. 3.2. Two main approaches have been proposed in the literature to address this kind of temporal boundaries (Baumann and Tillman 2004): (i) gathering emissions data within a certain period to complement the inventory, and accounting for the remaining materials after that period in a separate data category (Tillman et al. 1994), or (ii) including the landfill in the inventory until all the material is degraded (Finnveden et al. 1995).

To conclude, the system boundary is specified in the goal and scope definition of an LCA, as also explained in the ISO (2006a) and Curran (2017), but in this section, we have seen that it has several feasibility implications in the LCI modeling, which are still under debate.

2.4 Modeling Locations

Modeling locations in LCI models are interesting for a variety of reasons, both in the definition of the product system, in the LCI model, and in the impact assessment. Firstly, to build a realistic model, the location of a process delivering a product and the location of the process receiving that product needs to be identical, if the two processes are not linked with a transportation service: for instance, electricity from Norway cannot directly be used in Germany; it needs to be transported first. Secondly, processes and related flows can differ in different locations, especially when they depend on nature. For example, the same agricultural process might need a different amount of (or no) water at all in different regions due to different climatic conditions. Similarly, a photovoltaic cell will have different yields per year, depending on the altitude and the latitude. Thirdly, the same withdrawal of resources or release of emissions may have different impacts depending on the location, for example, on the availability of resources in the location or the status of air or water bodies. Emitting particulate matter has higher impacts in inner cities where more humans are exposed and withdrawing water in the arid region such as the Arabic peninsula has higher impacts than withdrawing the same amount in water-rich areas, such as the Netherlands. The required spatial detail and resolution depends on the scale of impacts, that is, if they are local, regional, or global.

To model locations in LCI, there is currently not one single, agreed-upon approach. Depending on the LCA databases and tools, different approaches are instead applied. As a basic approach to address locations, most LCI databases further classify flows into sub-compartments (e.g., the ecoinvent and the ILCD/PEF database). This allows for specifying (i) if emissions into air occur in high-population or low-population density area, which is useful to improve the characterization of human health-related flows; and (ii) if emissions into soils are located in agricultural, industrial, or forestry areas, or if water emissions end up into different types of water bodies (lake, groundwater, river, fossil water). This approach can be referred to as spatial archetypes (Mutel et al. 2018).

In addition, flows can be also regionalized or countrified, meaning that the database provides individual elementary flows for specific countries or regions, which are usually characterized using an ISO two-to-three letter code, possibly adapted and extended by different available databases (e.g., RER = rest of Europe, CH=China and RNA = rest of North America in ecoinvent). This approach can be adopted for and applied to processes. With this approach, flows can obviously be distinguished by country, and thus water withdrawal in Saudi Arabia can be distinguished from water withdrawal in the Netherlands. A drawback is that each flow is repeated for several locations and the dimensions of the database increase.

In the ILCD format, the flow is therefore not linked to a location, but instead only the exchange (i.e., the link between a flow and a process), meaning the flow, when it is input or output of a process, is linked to a location (Fig. 2.11).

This does not increase the number of flows in a database but is at present not yet supported by LCA software tools. Furthermore, a country or region, characterized with an ISO code, is not fully homogenous. Knowing that a given water withdrawal takes place in big countries, such as Russia, the US, and China, does not help much to understand the impact. Some LCA software systems integrate geographic information system (GIS) data or provide interfaces to GIS (at present, openLCA and brightway2). The use of GIS files makes the selection of the location more flexible. This improves the impact assessment, especially for those environmental impact categories focusing on the local scale, for which the country scale is too coarse. Those potentially more accurate inventories should be coupled with impact assessment methods able to deal with the same spatial scale (Frischknecht et al. 2018). For instance, when dealing with water, the water basin or watershed scale is usually adopted. For the AWARE water footprint method, a flow-based regionalization method with country-specific characterization factors for water scarcity is available, which is for big countries such as China typically not indicating the impact of water withdrawal at one given site (Boulay et al. 2018). In China, the water availability highly varies throughout the territories: The national average is about 43 m^3/m^3, meaning that China has about 43 times less available water remaining per area than

Fig. 2.11 Process, exchange, flow, and location, in the ILCD format

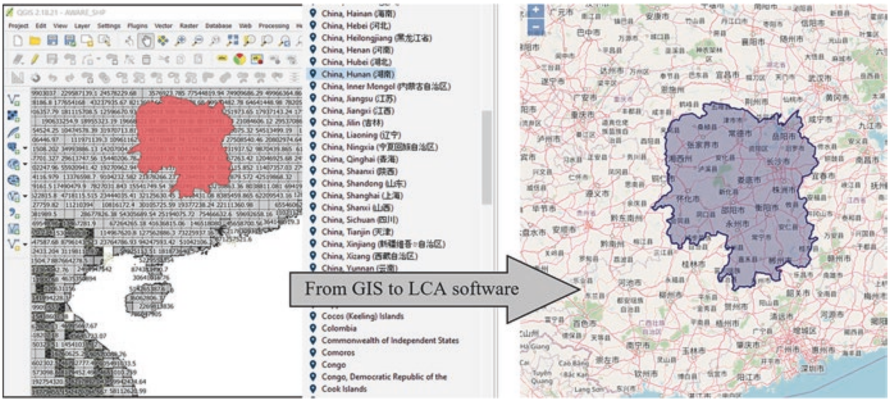

Fig. 2.12 Integration of GIS software (Q-GIS) and an LCA software (openLCA) to perform the geospatial-based regionalization. The left-side figure shows the south-eastern part of China (in gray) and the Hunan region highlighted in red; the grid shows information about water availability (from WaterGAp model, Flörke et al. 2013; Müller Schmied et al. 2014) in each single cell. The right-side figure shows the openLCA interface showing the same location (with OpenStreetMap, https://www.openstreetmap.org/, in the background)

the world average. This means that, if 0.5 m^3 water is used in China, the resulting impact will be about 22 m^3 (i.e., about 43 times higher than the input value). But China also has rainy regions, like Hunan (Fig. 2.12), where the average annual precipitation ranges between 1500 and 2000 millimeters (Hunan Gov. 2018). By integrating georeferenced watershed level characterization factors provided by the AWARE method and calculating the characterization factors specific for the Hunan province, a value of about 0.4 m^3/m^3 is instead obtained, and the impacts caused by the use of 0.5 m^3 of water instead becomes 0.2 m^3, about 100 times lower than the previous estimation.

With GIS integration or interfaces, there is no limit to the timely resolution of the inventory and of the impact assessment models, but it is evident that this can lead to extremely large models. As often in modeling, there are trade-offs between model sophistication and accuracy and effort spent.

Typically, in most LCA studies, specific geographical locations will be modeled for the foreground system only, while for the background system, generic locations might suffice, as they are provided in databases. These generic locations, however, can also be modeled following a site-specific approach, in case of large water power plants for example (Ribeiro and Anderi da Silva 2010).

In summary, modeling geographical location in LCA is still under development and the comparability and reproducibility of regionalized LCAs are not facilitated by any standards yet (Frischknecht et al. 2018). Nevertheless, the opportunities for the near future seem promising especially given the great availability of spatially explicit information available.

2.5 When Can a Process Dataset be Considered Complete?

As introduced in Chap. 1, an LCA tries to provide a comprehensive view of the environmental impact related to a product or service, in line with a specified goal and scope; the smallest modeling entity of a life cycle inventory model are process datasets. This directly raises the question as to when a process dataset can be considered complete. The requirement to provide complete or almost complete datasets can be found in almost all recent data quality management systems in LCA, see Chap. 5. So, under which conditions can a dataset be considered "complete"? There are three aspects to consider.

Firstly, the goal and scope specified for the dataset, the entire database, or also for the study determine the intended use and the impact categories and impact methods a dataset is supposed to support. For example, in one of the very early LCA studies, ozone depletion could not be considered as category, since in the data collection, ozone-depleting substances were not covered: *"the present LCI results do not allow for a consideration of the category Ozone Depletion, since in data collection, ozone depleting substances […] were disregarded"* (Schmitz 1995, p A12).[2] In addition to the supported LCIA methods, goal and scope also specified the nomenclature and thereby the structure and detail for the flows, to, for example, understand whether a dust emission should be called "dust," "fine dust," "particles," or be distinguished into "PM10," "PM5," and "PM2.5," to name just some examples.

Second, completeness is to be assessed based on the set of flows provided for the dataset; does it include, for example, CFC emissions, if these are present, or have they been skipped? Ideally, the dataset contains all flows occurring in reality, following the specified nomenclature.

Third, a dataset should have an even mass and energy balance, given that the basic LCI model does not foresee any stocks to be "stored" in the dataset: all masses that enter the process need also leave the process.

While it is easy to spot usage of inappropriate nomenclature in a dataset, it is much more difficult to see whether all resource needs and emissions have been listed, and even more whether the amounts are fitting to the real process.

3 Extensions of the Basic LCI Model

3.1 Modeling Multifunctionality

In the simplest case, each process produces one product, which represents the purpose or function of the process, meaning that the process is happening because there is demand for this product (e.g., a photovoltaic cell is built and installed because electricity is required). This latter links this process to other processes in the product

[2] Translated to English by the authors

system, making it the final product (i.e., quantitative reference) of the product system under study. Quite often, however, a process is creating more than one product (Fig. 2.13), and thus has several functions. For instance, in the soy biodiesel process shown in Fig. 2.5, there are two products: the main product soy biodiesel and the byproduct glycerin.

How can inventory models deal with such multifunctionality? This is one of the "classic" questions in LCI modeling (Russell et al. 2005). The ISO recommends a stepwise procedure (ISO 2006b). Firstly, to avoid allocation:

1. Dividing each unit process into subprocesses (each one referred to one coproduct) and gathering the additionally required environmental burden data.
2. Expanding the product system boundaries to include additional functions related to the coproducts.

If these options are not possible, then allocation is recommended:

3. Distributing ("allocating") the environmental burdens of each product based on their underlying physical relationships, i.e. their mass, or energy content for example.
4. If allocation based on physical relationships cannot be done, then this allocation of the environmental burdens of each product needs to be done based on other rules; often, economic relationships, i.e. the price of the products, is then used.

When developing an LCI model, the three approaches reported in the ISO standard have a different practical implementation. In the first case, system subdivision, additional effort is required to refine data collection while focusing only on the product under study, which in real systems is often not feasible (Fig. 2.14). Additionally, to have accurate information and results, the subprocesses should be physically and economically independent (Ekvall and Finnveden 2001). In the end, subdivision is possible in cases where the two independent processes have been lumped together somewhat thoughtlessly in an initial model. An example: A soybean farmer may produce soybeans and also wheat, but for 1 year in different fields; for an LCI model, a combined process could be created that produces both wheat and soybeans, and thus is a multifunctional process. Instead, though, two processes

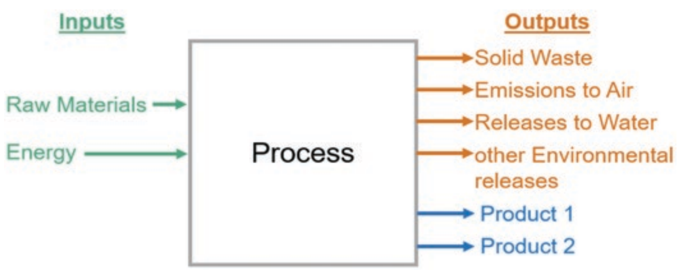

Fig. 2.13 Production system producing two usable output products

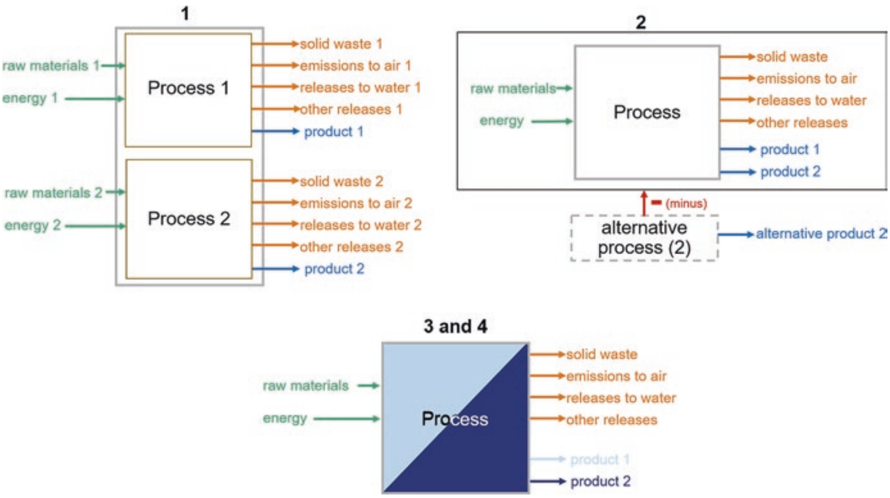

Fig. 2.14 Dealing with multifunctionality: system subdivision (1), system expansion (2), avoided burden approach), and allocation (3 and 4)

could be created as well, the one producing wheat, the other soybean, without the need to apply allocation or system expansion.

Secondly, system expansion can be performed via system enlargement or the avoided burden approach (Azapagic and Clift 1999), which consists of either adding or subtracting (Fig. 2.14) the environmental burden of an *alternative production process* (Reap et al. 2008). The alternative process provides then only the one byproduct from the initial process, and its purpose is to basically get rid of the byproduct. Including alternative production into the LCA model is not always an easy task since it leads to a larger, more complicated model that requires more data (Curran et al. 2005). Additionally, (i) an alternative process may not exist, or (ii) there may exist more than one alternative process, and (iii) the required inventory for the alternative process may not be accessible or reliable (Azapagic and Clift 1999).

Thirdly, the allocation is an option (Fig. 2.14); it is sometimes considered as one of the most controversial issues in LCA (Rebitzer et al. 2004; Reap et al. 2008). As previously said, different allocation rules exist (e.g., physical or economic), and it typically cannot be stated that one single method of these is best or provides a generally acceptable solution (Curran 2007). Dealing with multifunctionality thus remains a matter of choice in the approach and in the method within each approach. The decision for one or the other way to deal with multifunctionality should be made in the goal and scope definition since it might have a strong implication on the created LCI. For a deeper discussion about the multifunctionality problem, see Chap. 4.

3.2 Modeling Time

Real-life production processes, and consequently product life cycles, happen in and vary over time. Each single process in a life cycle is characterized by a certain temporal dimension (e.g., duration), for example, in the production steps and in the use phase. The LCI attempts to describe a dynamic, time-dependent, and successive-in-time technosphere, both in processes and supply-chains (Shimako 2017). However, the temporal dimension is normally not considered in the basic LCI model, which is rather agnostic about time. LCI is indeed implemented by assuming an "infinite" flow of products, from one process to the next one in a given life cycle, in consecutive periods of time (Fig. 2.15) and by assuming that all the involved technologies were to remain the same (i.e., steady-state technological relations between a process and its inputs and output flows) (Ciroth et al. 2008).

In practice, LCI results consist of absolute quantities (e.g., kg) and, despite their name, not of physical flows in a strict sense (i.e., kg year^{-1}). Emissions, consumed resources, and intermediate technological flows are expressed without a time indication. Nevertheless, time affects LCI modeling in a variety of ways. Here below, some examples are presented. Firstly, time affects different life cycle stages: (i) during the use phase, some products, such as vehicles and buildings, need maintenance activities. The models generally assume maintenance interventions at predefined time intervals over the (assumed) life span of the product; (iii) in real processes, a storage phase can occur between an input and an output, and in a

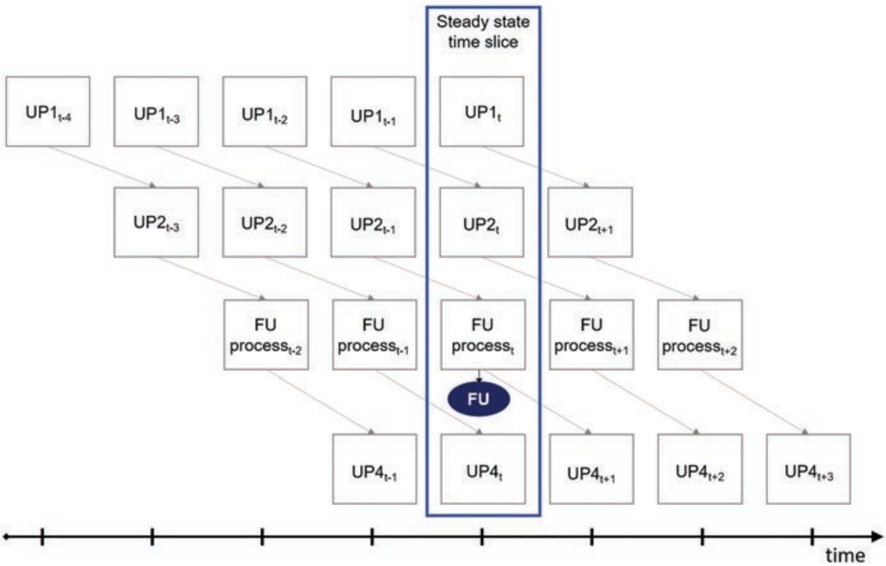

Fig. 2.15 Steady-state technological relations between a process and its inputs and output flows and steady state time slice analyzed with LCA (Adapted from Ciroth et al. 2008). (UP = Unit Process; FU = Functional Unit; t = unit of time)

warehouse, stock can decrease or increase. In balance sheet equations in engineering (e.g., Schütt et al. 1990), this effect is described with a storage term, while in LCI modeling this term is simply ignored. With this simplification, the input must always even output, and every flow must in principle represent a steady-state, where it is not possible to have ever-increasing or ever-decreasing stocks; (iii) products may last and be used for a very long time (e.g., buildings and infrastructures), so long that end-of-life options are hard to imagine, both in technological and regulatory terms.

Secondly, in seasonal processes, such as in agricultural systems (Notarnicola et al. 2017), input and output flows vary within a year and this is often not accounted for in LCA (Bessou et al. 2013). The consumption or acquisition of certain resources and emissions can have different impacts at different times and in different locations. For example, a certain amount of water consumed during the dry season has higher or at least different impacts than the same amount consumed in the wet season (Boulay et al. 2015a, b).

Thirdly, different technologies studied in LCA might have reached different technological maturity. In the early days of LCA, the objects of study were often products that were mature in the sense that they had been produced and used in society for a long time, such as steel, concrete, milk, and ketchup. Today, many of these mature products have been thoroughly assessed and the focus has often turned to the continuously ongoing technology development. New materials, products, and technologies are being researched and developed each day. The benefit of assessing technologies at such and an early stage of technological development is that much of their design is still open to alterations at modest costs and efforts. An example of an emerging technology could be an electric car, to be compared by the current mature personal transportation technology, which would be a car with a combustion engine. Such comparisons can be facilitated by envisioning the emerging technology in the future, more mature state, which might include both upscaled production processes and altered background systems (Hillman and Sanden 2008). This approach can be referred to as prospective or ex-ante LCA (Villares et al. 2017; Arvidsson et al. 2018; Cucurachi et al. 2018). There is yet no standardized method for estimating the future emissions and resource use related to an emerging technology. Rather, a number of different approaches are being tested. One approach is to apply scenarios reflecting future, large-scale production (Walser et al. 2011; Arvidsson and Molander 2017). Laboratory-scale production is typically characterized by low energy efficiency inefficient use of materials, such as solvents. At larger-scale production, on the contrary, saving energy and materials is of high environmental importance and might also bring benefits in terms of reduced costs. Another suggested approach is to apply technological learning curves for future upscaling in prospective LCI modeling (Bergesen and Suh 2016).

Additionally, within a life cycle, consumption of resources and emissions at different stages can occur at different times, sometimes with time lags of decades or even centuries. This time lag raises issues related to intergenerational fairness and equity-related to current and future impacts. Methodologically speaking, by assigning the same characterization factors to short-term and long-term emissions, the impact of the latter might be overestimated. Furthermore, "releasing a big amount of pollutant instantaneously generally does not have the same impact as releasing the same amount of pollutant at a small rate over several years" (Levasseur et al. 2010).

A simple time differentiation in common LCI has started at the beginning of 2000 (Hellweg and Frischknecht 2004), in particular, with a distinction between long-term and short-term emissions. In practical terms, the basic LCI model assumes the following (ecoinvent 2018):

– Short-term emissions (e.g., occurring with a time horizon of 100 years in the ecoinvent database) are included in the modeling and are all assumed to occur at the beginning of the analyzed horizon and therefore aggregated; long-term emissions (e.g., after 100 years) are usually disregarded; this is done to prevent distorting assumptions in case of continuous emission over very long time spans, which would lead to an infinitively high emission which could distort any inventory result in case of time neglect (e.g., the case of emissions from landfill sites, or Radon-222 emissions from Uranium extraction and milling).

A more complete temporal resolution over the entire LCI is also possible, as proposed in dynamic LCA (Levasseur et al. 2010). An inventory where temporal differences have been considered can be referred to as a temporally-differentiated life cycle inventory (Beloin-Saint-Pierre et al. 2017). Clearly, refining and increasing the accuracy of methods to describe flow dynamics would improve LCI modeling, but integration with impact assessment methods that consider this dynamic is needed. Nowadays, even if different methods have been proposed in the literature (especially in the case of climate change, see Brandão et al. 2013 for a review), the definition of characterization factors is based on annual averages, without really making a distinction of time horizons. The ecoinvent database, for instance, provides two options: (i) attributing the same characterization factors to both short-term and long-term emissions, leading to an over-estimation of the impacts; (ii) attributing no characterization factors to the long-term emission, leading to an under-estimation of the impacts.

Conceptually, though, integrating time in the inventory is not complicated. Processes need a start and an end time, and when building the life cycle, the software or practitioner needs to understand which processes start at which point in time. As a practical example for long-living goods, temporally differentiated inventory models and full case studies for train components were published around 2000 (Ciroth et al. 2003): in the inventory of a train component with a lifetime of 30 years, a series of processes happen, with daily cleaning of the train to refurbishment and predictive maintenance and occasional accidents, where some prevent the train from operating (Fig. 2.16).

Discounting methods have been developed specifically for LCI modeling to take into account the time effects of biogenic carbon emissions, meaning carbon-related input and output flows involving biomass. When biomass enters a production process (e.g., wooden furniture or building construction, or production of food), the resulting CO_2 can be considered as a negative emission in the LCI. Usually, when

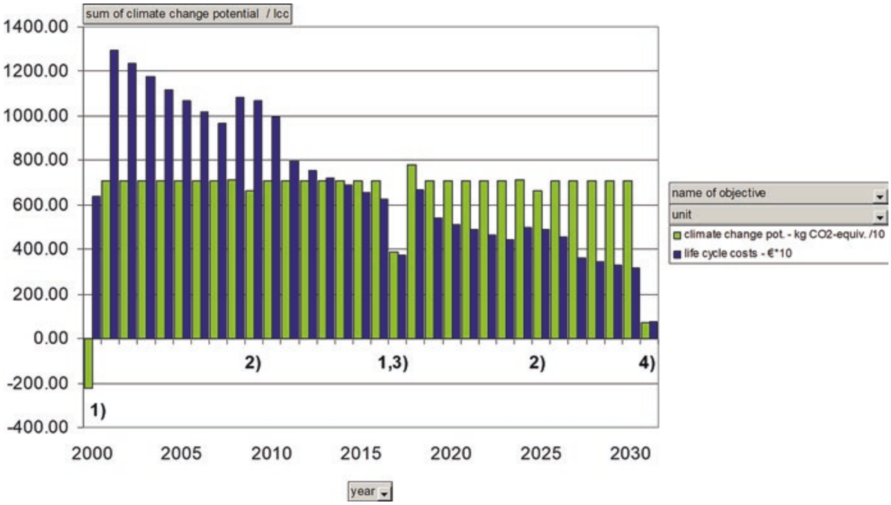

Fig. 2.16 Life cycle costing and climate change (in kg CO_2 eq) results for a wooden floor in a train carriage, operated for 30 years: (1) negative potential due to incorporated CO_2; (2) revision of the train; (3) modernization and reproduction of the floor; (4) disposal (waste incineration plant). Costs are discounted by 5% (Jensen and Remmen 2005, p. 84)

considering carbon uptake, a distinction is made between "short-living" goods, such as food, and long-living ones, such as furniture. The negative emissions of short-lived goods are typically not considered, since the CO_2 will be emitted again, for example, after the food is consumed, and a negative contribution to climate change for food at the point-of-sale might be misleading. For long-living goods, negative emissions are considered, since the goods "preserve" the captured CO_2. Altogether, different options in dealing with biogenic carbon uptake, storage and fixation, and removal are present in the literature:

- Biogenic carbon can be excluded from the inventory model entirely, that is, CO_2 fixation by vegetation is not considered, nor are downstream biogenic CO_2 emissions (e.g., in the case of food, or incineration of paper) (Milà i Canals 2007).
- Biogenic carbon can be considered together with the fossil carbon, that is, consider CO_2 fixation by vegetation as a negative emission and then account for the emission wherever it occurs (e.g., in waste treatment) (Milà i Canals 2007).
- Biogenic carbon can be distinguished from fossil carbon, and considered and reported separately from fossil CO_2, as required by the specific ISO for carbon footprint (ISO 2018).

Referring strictly to time, according to the ISO 14067 (ISO 2018), all the emissions (and removals) occurring over the life cycle "shall" be included without the effect of timing. This means that emissions arising from the pre-use stages, the use phase, and the end of life must be considered as single releases at the beginning of the time horizon chosen (Fig. 2.17).

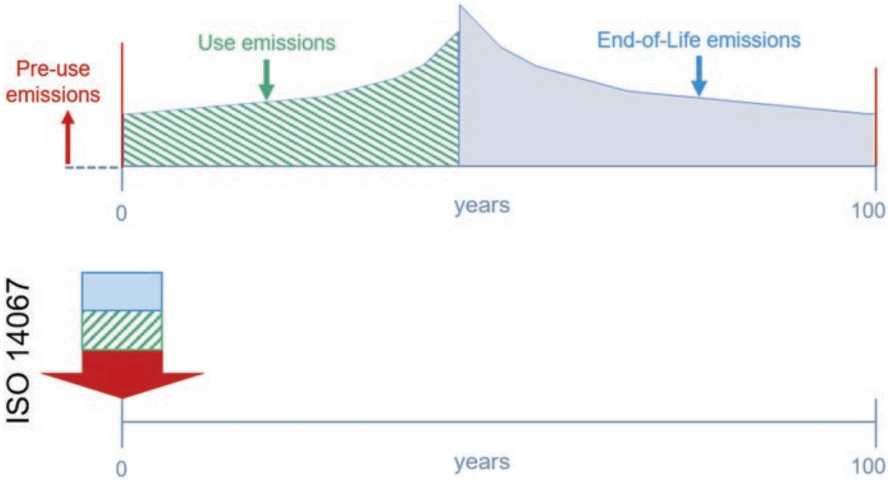

Fig. 2.17 Delayed greenhouse gas emissions: reality vs the ISO 14067 approach

The importance of considering the temporal dimension in LCA has been highlighted by several studies (e.g., (Hillman and Sanden 2008; Reap et al. 2008; Finnveden et al. 2009; Collinge et al. 2013)). Time can be integrated into LCI through different methodological modeling approaches (e.g., from the area of financial accounting; see Ciroth et al. (2008) for a brief review of the types of models), or can be included as part of uncertainty analysis (Huijbregts et al. 2001; Stasinopoulos et al. 2012; Collinge et al. 2013). Recently, also in line with more powerful modeling capabilities and better data availability, there is increased interest in integrating time in LCI models and create dynamic LCIs, introducing temporal parameters in processes (Tiruta-Barna et al. 2016; Beloin-Saint-Pierre et al. 2017; Shimako 2017).

3.3 Low Probability Flows of High Impact, Unknown Mechanisms

The basic LCI model is deterministic. Flows are modeled at an often-unspecified time and location, but with certainty. Some practitioners and databases add uncertainty information to flows, but this is done mainly to address the reliability and data quality of the information rather than the probability of the flow actually occurring at all. Only deterministic, fully certain flows are captured in an LCI model, while flows that occur with low probability are excluded. If these flows lead to impacts, the impact can be called risk following the classic definition in risk assessment, where risk equals probability time impact (Fig. 2.18).

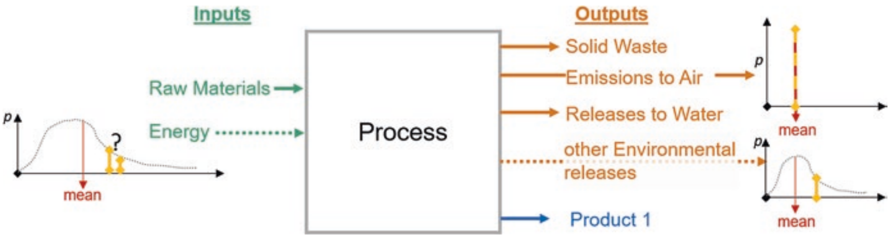

Fig. 2.18 Modeling risk in LCI: dashed arrows show uncertain flows (i.e., energy input and other environmental releases) characterized by a certain probability density function with a given mean (red arrow); the yellow lines represent the possible amount of flow occurring in the real system: these values can be far from the average for uncertain flows, while they are equal to the average for deterministic flows (e.g., for emissions to water)

In many real-life situations, flows and their subsequent impacts occur only with low probability. They are unplanned and indeterministic. Some example flows include:

– Nuclear power plants and treatment of radioactive substances with a certain, low, probability for emission of radioactive substances
– Oil leakages from oil extraction plants or during transportation via oil tankers
– Littering of plastic (e.g., determination of plastic waste path, from a source to a destination, e.g., a certain marine area)

If these flows are linked to a high impact, then this calls for an assessment of the entire risk of the process and related flow occurrence, where the risk is, as in risk assessment, obtained as a product of occurrence probability and damage, that is, environmental impact.

Slightly different cases are flows, where the behavior, their derivatives, and metabolites in the environment are simply not fully known, and accordingly, the mechanisms of these flows are currently not fully known. Examples include genetically modified organisms and nanoproducts. Nanoproducts emissions are nowadays in LCA addressed via a deterministic model, following an international workshop recommendation, summarized in Klöpffer et al. (2007):

> *"LCA gives a more holistic picture of the environmental impacts of products than does RA [Risk Assessment] alone. Furthermore, it allows the identification of the life cycle stages, the stage at which major environmental impacts may occur, and the potential risk of exposure for different people along the product-transformation chain. LCA provides very useful indications for improvement and potential impact minimization."*

Even though this earlier source also highlights the need for more information on inventory and impacts, LCA is identified as the main tool. Today, authors seem more cautious and increasingly stress that qualitative and risk-based information would be needed to complement the analysis (e.g., Curran 2015, p 49):

> *"If LCA is used exclusively to assess the environmental impact of a nanoproduct, it will adequately capture the issues related to resource management and climate change issues […]. However, shortcomings may arise because models for the underlying characterization*

of impacts to human and ecosystem health are underdeveloped […]. Thus, the incorporation of a modified risk-based health impact model accounting for chemical factors under site-specific conditions into the LCA framework would achieve a maximum understanding of the impacts of a nanoproduct to guide decisions […]".

In LCI, specific events can of course be addressed via sensitivity analysis, in which the event is described as a variation of flows (e.g., increase or decrease of a certain type of emissions). This variation can even be characterized with a certain probability. However, this approach is used to address rather few additional modeling options than full inventories that are not deterministic, and even less to model chains of consequences with probability. For these, other tools "outside" the usual LCA modeling domain exist. For instance, Bayesian networks (BN) used for failure mode effect analyses (FMEA) and risk assessment are found to be able to address questions such as the reliability of technical plants up to nuclear power plants adequately (although not as only tool to be used in practice) (Fig. 2.19).

This short discussion shows that LCI modeling is not able to fully capture the impacts of certain products. A basic LCI model has difficulties to address (i) flows that are highly uncertain in occurrence, but still important due to a potentially high impact, and (ii) flows that cause impacts that are uncertain. Both challenge the understanding that an LCA constitutes a comprehensive assessment of the potential impacts of a product over its life cycle. The second difficulty might mainly be an

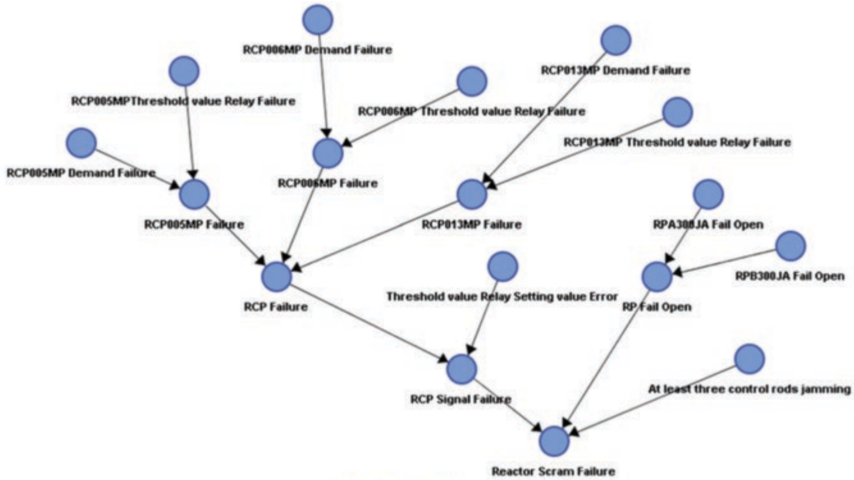

Table 1 Probability of root nodes

Root nodes	Probability	Root nodes	Probability
RCP005MP Demand Failure	6.13E-03	RCP013MP Threshold value Relay Failure	1.00E-04
RCP005MPThreshold value Relay Failure	1.00E-04	Threshold value Relay Setting value Error	1.50E-04
RCP006MP Demand Failure	6.13E-03	RPA300JA Fail Open	3.20E-04
RCP006MP Threshold value Relay Failure	1.00E-04	RPB300JA Fail Open	3.20E-04
RCP013MP Demand Failure	6.13E-03	At least three control rods jamming	1.00E-04

Fig. 2.19 A simple failure mode effect analysis for the Daya Bay nuclear power plant in Country X (Yu et al. 2003)

issue for the impact assessment phase, although the uncertain impacts of entities released to the environment may also call for a different modeling of their flows. The first difficulty is usually addressed in risk assessment, albeit not in a life cycle perspective. The inclusion of risk assessment within LCA and LCA integration with risk assessment has been debating in the last decades. Risk assessment aims at (i) identifying hazardous events that can affect some endpoint (e.g., humans, and the environment), (ii) assessing the likelihood of these events, and (iii) its potential consequences (Kaplan and Garrick 1981). First attempts to include risk in LCA were focused on chemical pollutant toxicity (human and eco) and took inspiration from knowledge in chemical risk assessment (e.g., Guinée and Heijungs 1993; Keller et al. 1998). Differences, synergies, and potential integration between these two disciplines have been extensively discussed in the literature (e.g., Olsen et al. 2001; Cowell et al. 2002; Hofstetter et al. 2002; Bare 2006). According to Harder et al. (2015), most recent studies dealing with integration between risk assessment and LCA can be classified into three main clusters: (i) site-dependent assessments, which start from environmental input-output analysis aim at assessing spatially differentiated human health risks; (ii) applications of life cycle thinking in risk assessment, which implies an enlargement of the risk assessment scope to include the entire life cycle of products; and (iii) trade-off between local and global effects, tackling the issue of burden shifting when focusing on specific contexts.

4 Life Cycle Modeling Conventions

4.1 Modeling Transport Services

Transportation is involved in most LCA studies. Modeling transportation services includes the production of vehicles, the use phase that usually involves fuel consumption and related emissions, but also construction and maintenance of transport infrastructures (e.g., roads and rails), as well as the end of life of vehicles. Transportation models involve crucial assumptions on the life span of vehicles, and average load, and average traveling or working conditions (i.e., in terms of emissions). Additionally, modeling the *use* of roads and other infrastructures by each type of vehicle requires extensive data collection, which is seldom regularly updated (Spielmann et al. 2007).

Two main LCI approaches have been developed to model transportation. The first one considers both the transported mass (e.g., a ton) and traveled distance (e.g., km). The environmental exchanges are related to the reference unit of one ton-kilometer (or kilogram-kilometer, or 1 kg over 100 km), which is defined as the transport of 1 metric ton of goods by a certain transport service over 1 kilometer. The forward and return trips can be already included in the model, and the only data to be collected are transported mass and traveled distance (usually only one-way distance). The accuracy of the model can be increased through the use of parameters

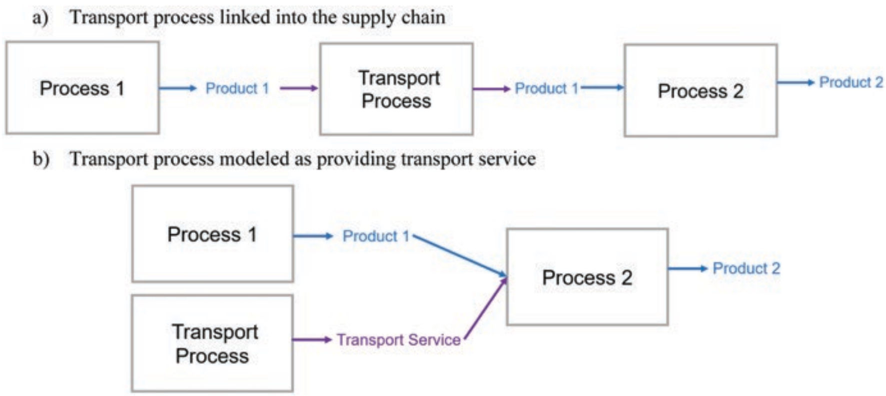

Fig. 2.20 Modeling generic transport processes as linked into the supply chain (**a**) or as a separate process providing transport service (**b**); principle example for transporting product 1 from process 1 to process 2 in a supply chain

such as level of utilization with respect to the maximum load capacity (i.e., ratio between actual load and maximum total payload), specification for forward and return trips (e.g., different loads or utilizations), and fuel characteristics (e.g., sulfur content and share of biogenic CO_2). This first model is applied to both cargo and passenger transportation.

A second modeling approach has been developed to describe passenger vehicles (i.e., cars). The functional unit is 1 vehicle-kilometer for passenger car processes. This means that the only information to be collected in the traveled distance and information about the traveled road categories, such as urban, rural, or highway, while the transported mass is not considered.

When creating the LCI model, transport can for one be modeled as service delivered to the process requesting transport, or it can be a separate process that links the process providing the product and the process "consuming" the product, so that the transport process is directly linked into the supply chain (Fig. 2.20).

Both approaches are common. Databases and software tools that request unique product names typically follow the "transport service" modeling approach, including the ecoinvent database, and SimaPro as LCA tool; the GaBi database follows the approach to link transport processes in the supply chain.

4.2 Modeling the Use Phase

Modeling the use phase in LCI is a challenging step. Usually, differently from the production process, a specific product can be used in several alternative ways. A laptop can be intensively and continuously used for working activities or just for a few times per week for leisure activities. For this reason, modeling the use phase often means developing expected average or typical consumption patterns that can highly deviate from the actual use of a single product by a single end-user. Therefore, several assumptions have to be adopted, such as lifespan of the product, number of uses, type and number of maintenance activities, as well as type and amount of energy used.

For instance, in the case of transportation, especially those requiring fuel, use phase impacts depend on assumptions regarding the lifespan of the vehicle, the average load, average fuel consumption per kilometer, and maintenance interventions (e.g., substitution of wheels and other components) (Spielmann et al. 2007). For food items, such as pasta, average cooking conditions need to be assumed. Usually, pasta is boiled for a given period of time in a given amount of water, which is heated with gas (in Italy). The time period, amount of water, and type of energy can change and the environmental impacts can therefore vary considerably (Ruini et al. 2013). Inventory processes in the use phase can be grouped into product-independent and product-dependent processes (European Commission 2018). *Product-independent processes* do not depend on the way a product is designed or distributed. They do not contribute to any differentiation between two products. For example, CO_2 emissions related to the electric grid mix, where electricity is used for boiling 1 liter of water used to prepare instant coffee is independent from the specific product design. *Product-dependent processes*, instead, are determined or influenced by the product design or use instructions and contribute to differentiation between two products. An example is the efficiency of different water boilers, that is, the consumption of more or less electricity for bringing 1 liter of water to the boiling point.

Often, modeling the use phase is related to the main function of a product, such as the electricity consumed during the utilization of electric and electronic tools (e.g., washing machine or laptop). Besides this approach, the *Delta approach* can be useful (European Commission 2018). This is applied when the use of one product influences the environmental impacts of another product, and it involves allocation. For instance, toner cartridges are not held responsible for the consumption of paper they print, but if remanufactured, the toner cartridge works less efficiently and causes more paper loss compared to an original cartridge, the additional paper loss should be allocated to the remanufactured cartridge (European Commission 2018).

The use phase can have a great contribution to the environmental impact of a whole life cycle, as for energy-using appliances (Throne-Holst et al. 2007), and the consumer behavior can significantly influence this contribution (Solli et al. 2009). This relevance was realized already in early LCA studies (e.g., Eberle and Franze 1998; Jönsson 1999). For these reasons, in current LCA practice, the use phase is typically modeled explicitly (Polizzi di Sorrentino et al. 2016). At the same time, the use phase is often not fully under the influence of the producer (e.g., how long a user takes a shower and how they set the water temperature for the shower), and it is not even easy to know since users will usually not document their behavior easily and in an accessible way. Recently, the need to go beyond average use patterns and taking into account interindividual behavioral variation while modeling different usage scenarios has been highlighted by several studies (see Polizzi di Sorrentino et al. 2016 for a review).

4.3 Modeling End of Life

Waste can occur at different points of a product system, either during the production, the distribution, or after the use phase. Modeling the waste and end of life is therefore a crucial element in LCI, but it is challenged by the dependence on waste source (which also influences the type of waste, e.g., industry or households), geographical origin, transportation (de Beaufort-Langeveld et al. 2003), waste treatment, and technology, as well as by the time lag between production and end of life (e.g., in the case of buildings). This makes the identification and description of waste treatment options more difficult.

There are two ways of modeling waste flows and treatment in LCA software (Di Noi et al. 2017, see Fig. 2.19). The first approach considers the waste treatment as a "service" for the process to eliminate the product. It is also called *opposite direction approach*, since waste treatment is added as input into the process preceding the waste treatment itself or the main process (i.e., process that delivers the reference flow or final product), and waste flow is added as a negative input, which mathematically means that it is an output. The same flow is an output in the waste treatment process.

The second approach follows the more natural direction of flows from one process to the other, meaning that the waste is an output from the production or use phase from which it is generated. In this case, a specific type of flow (waste flow) is defined and inserted as output in the waste-producing process, as well as an input in the waste treatment process.

The modeling described above refers to waste and needs to solve the problem that waste treatment processes offer a service that is "opposite" to normal production processes and thus somewhat contradict normal LCA process modeling. Normal processes produce products that are providing a value and deliver these to other processes. The product is the output of these processes. Waste treatment processes provide benefits by *accepting* waste (on the input of the process). Both the "opposite direction" and the "flow logic" approach solve this by reversing the direction of the waste flow.

Modeling recyclates, that is, substances that have been created or produced and are now used a second time, is also part of the end-of-life modeling but poses, in addition, the question of how to distribute, or allocate, the burdens over the two (or even more) life cycles. According to Baumann and Tillman (2004), some recyclate allocation methods have been framed around *fairness*, meaning which product or process is responsible for raw material extraction, waste production, and recycling. Other more *change-oriented methods* consider what would happen if the recycling system is changed. The first group includes:

(i) The *cut-off* method that assigns only direct impacts to a given product (e.g., extraction of virgin material is allocated only to the first product) and does not require data from outside the investigated life cycle.

(ii) Allocation based on the *relative loss of quality* in subsequent recycling. This method allocates environmental burdens according to the quality of the material, which is supposed to gradually decrease from recycling to recycling.

(iii) Waste is seen as a consequence of raw material extraction and thus allocated to the *first* production process that is responsible for the raw material extraction. This method promotes the *use* of recycled material.

(iv) Waste can also be allocated to the process that does not recycle, while to the processes that ensure waste recycling only the environmental burden caused by recycling is assigned. This method gives *incentives to produce recyclable products*.

The second group of methods is based on change-oriented arguments. System expansion can be the suitable approach, but given the additional data requirements and uncertainties, allocation methods that are approximations of the system expansion have been developed:

(v) *Closed-loop recycling approximation* uniformly allocates environmental burdens of raw material extraction, waste production, and recycling to all the processes involving these flows. This approximation is suitable for materials that do not lose quality when recycled and can therefore replace virgin materials.

(vi) For materials that lose quality during recycling and cannot be easily used in the same product, the closed-loop approximation is less suitable, and an alternative method is the *50–50 method* (Ekvall 1994). It assigns the burdens due to raw material extraction and waste treatment to the first and last product in the overall system (i.e., composed by the different product systems connected via waste flows) in equal proportions, and allocate the recycling process to 50% to the product upstream and 50% to the product downstream the recycling itself.

In the last years, further methods have been developed, including the Circular Footprint Formula (CFF) developed within the Product Environmental Footprint (PEF) project by the European Commission (European Commission 2018). The first version of the formula was presented in 2013, then referred to as the End-of-Life Formula. The CFF formula includes specification for material (virgin and recycled), energy (in case of energy recovery from waste), and disposal (Fig. 2.21). The general formula considers different possible material origins (i.e., virgin or recycled), waste treatment and their efficiencies, and quality of materials.

Material $$(1 - R_1)E_V + R_1 \times \left(AE_{recycled} + (1 - A)E_V \times \frac{Q_{Sin}}{Q_p}\right) + (1 - A)R_2 \times \left(E_{recyclingEoL} - E_V^* \times \frac{Q_{Sout}}{Q_p}\right)$$

Energy $$(1 - B)R_3 \times (E_{ER} - LHV \times X_{ER,heat} \times E_{SE,heat} - LHV \times X_{ER,elec} \times E_{SE,elec})$$

Disposal $$(1 - R_2 - R_3) \times E_D$$

Fig. 2.21 PEF Circular Footprint Formula

***A*: allocation factor of burdens and credits between supplier and user of recycled materials; ***B*: allocation factor of energy recovery processes: it applies both to burdens and credits; Q_{sin}, Q_{sout}, Q_p: quality of the ingoing secondary material, of the outgoing secondary material, and of the virgin material; **R_1:** proportion of material in the input to the production that has been recycled from a previous system; **R_2:** proportion of the material in the product that will be recycled (or reused) in a subsequent system; **R_3:** proportion of the material in the product that is used for energy recovery at end of life; **$E_{recycled}$:** specific emissions and resources consumed arising from the recycling process of the recycled (reused) material; $E_{recyclingEoL}$: specific emissions and resources consumed arising from the recycling process at end of life; E_v, **E^*_v:** specific emissions and resources consumed arising from the acquisition and preprocessing of virgin material, and of virgin material assumed to be substituted by recyclable materials; **E_{ER}:** specific emissions and resources consumed arising from the energy recovery process; $E_{SE,heat}$ and **$E_{SE,elec}$:** specific emissions and resources consumed that would have arisen from the specific substituted energy source; **E_D:** specific emissions and resources consumed arising from disposal of waste material at the end of life of the analyzed product, without energy recovery; $X_{ER,heat}$ and **$X_{ER,elec}$:** efficiency of the energy recovery process for both heat and electricity; **LHV:** lower heating value of the material in the product that is used for energy recovery.

The debate and research on how to tackle recycling materials are still ongoing and there is no consensus on a single approach. This topic is crucial when LCA is used, especially within the context of circular economy (Dieterle et al. 2018).

5 Conclusion

The basic LCI model is rather simple but at the same time proven to be very useful and successful in the last decades. Setting the functional unit, choosing an approach for modeling causality, setting system boundaries, and modeling locations are important aspects of the basic LCI model. The model can, however, be extended to become more realistic and to cover also more complicated production and service processes, as shown by the extensions described in this chapter. Modeling multi-functionality, time, and accidents are examples of such extensions. Whereas multi-functionality is commonly modeled through system expansion, allocation and substation in current LCA practice, the explicit consideration of time, and particularly the consideration of accidents are less common. These extensions might become more frequently applied in the future, possibly along with other extensions. If they become more frequent in the future, generic conventions for their modeling might emerge in a similar way as it already has for the modeling of certain often-occurring processes, such as transport services, the use phase, and end of life.

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